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Review and Categorization of Saltation, Suspension, and Resuspension Models

By W. J. Smith II,* F. W. Whicker,† and H. R. Meyer‡

Abstract: *Fifteen mathematical models for estimating the entrainment of particles by wind are reviewed and categorized to help investigators more easily match the requirements of their application to the capabilities and data requirements of the available techniques. A description of the origin and intended application of each model and a brief review of the formulation and parameterization are given. Important assumptions, limitations, and desirable features of the techniques are noted. A table outlines the suitable applications, available environmental and climatic variables, and the type of estimate provided by each model.*

The entrainment and transport of particles by wind has received considerable attention. A large body of literature is available, ranging from the resuspension of radioactive fallout particles to the wind erosion of agricultural soil. This wide range of models complicates the task of selecting an entrainment formulation suitable for a given purpose. This article gives information and background on available models so the reader may limit the number of techniques that must be pursued in more detail. The citation of references is limited to one or two important sources for each technique, which should provide a reasonable entry point to the appropriate literature.

Definitions follow for some fundamental terms used here. *Entrainment* is used to mean the general pickup and movement of particles by the wind. *Suspension* applies to particles (generally <50 μm in diameter) kept airborne for long distances by the force of the wind. *Resuspension* applies to suspension of particles that have been previously airborne and deposited and whose characteristics

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may be different from the soil. *Saltation* refers to the leaping or bounding motion of particles lifted by the wind that are too heavy to be held aloft and fall quickly back to earth. Saltation is generally the mode of movement for particles of 80 to 1000 μm in diameter. *Surface creep* describes the rolling or sliding movement of particles generally >1000 μm in diameter and too heavy to be lifted by the wind. *Respirable* defines a limited range of particle sizes that, when suspended in air, will enter and deposit in the human respiratory tract. The respirable sizes are generally defined as <7 μm in diameter. *Erodible* particle sizes, as defined by agricultural soil scientists, are those <840 μm in diameter. *Nonerodible* soil particles are defined as those >840 μm in diameter.

This article is intended as a reference for those interested in available models for particle entrainment by wind. Dust raised by mechanical disturbances or fire is not considered. As far as possible, we have tried to present the concepts and formulations underlying each model without judging which are "better" models and which are "poorer." Such judgment falls in the realm of model evaluation, and that task is highly dependent on the specific problem being considered. The model best suited for giving average concentrations of resuspended fallout particles above a Utah pasture is not likely to also be the best for estimating the mass flux of suspended dust from a rocky ore stockpile during a particular wind storm. We cannot anticipate the specific problems of each reader, of course, so some pertinent questions will go unanswered by our discussions. We hope, however, that the core of each model is presented in a fashion that will lead the reader to techniques useful for a particular application.

For brevity, we do not present values for many of the empirical parameters of the different models. Generally, the range of values may be expected to be large and highly dependent on the conditions at each individual site. Investigation of parameter values is left to the researcher to encourage the development of site-specific values rather than the use of "ball park" or "default" values. A useful review of data and parameter values has recently been published by Sehmel.[1]

For this review, model presentation is ordered such that the number of new parameters introduced for each successive model is kept small; if one model introduces information needed in a

second one, it precedes the second model. We could not adhere to this goal in every case, but we feel it gives a sense of progression and development to this group of interrelated modeling approaches.

REVIEW OF AVAILABLE MODELS

We have selected descriptive names for several models that may not always correspond with terminology used in other places. Each model or technique is discussed according to the following general scheme: (1) origin and intended application; (2) formulation and parameterization; and (3) assumptions, limitations, and presence of desirable features.

Mass Loading

The mass loading approach is perhaps the simplest method of estimating airborne radioactivity originating from the radioactive contamination of surface soil. It was proposed originally as an estimator of the resuspension of fallout particles from nuclear weapons testing. For more detailed discussions see Refs. 2–4.

Assuming continual entrainment and depletion of particles in the airstream, the dust loading of the air at a given point is indicative of the gross balance between the two processes. The airborne dust is assumed to originate from the suspendible fraction of the upwind surface soil. The dust loading M (mass/volume) is used to calculate the airborne radioactivity C_a (radioactivity/volume) expected over a contaminated area

$$C_a = MC_r \tag{1}$$

The radioactivity associated with the suspendible fraction of the surface soil C_r (radioactivity/mass) must be measured. Measurements of the atmospheric dust loading M are commonly time averaged. They may be averaged over space as well and may provide particle-size distribution information. When particle-size data are available for both the dust loading and the radioactivity in suspendible soil, Eq. 1 may be rewritten

$$C_a = \sum_{i=1}^{n} C_{ai} = \sum_{i=1}^{n} M_i C_{ri} \tag{2}$$

where the subscript i indexes the n particle-size classes.

Two important assumptions underlie this simple formulation. First, the requirement that the contaminated area be large is fundamental, because the available mass loading data represent the average condition for areas large enough that suspension and depletion of dust in the airstream can equilibrate.[3,4] Second, estimation of the radioactivity associated with the suspendible soil fraction requires reasonable uniformity of the contaminant in the contaminated area.

The mass loading approach is useful in situations for which suitable dust loading measurements are available. Because no environmental parameters such as soil moisture content or wind speed are incorporated into this model, care must be taken that the conditions at a given site are properly represented by the average conditions during the measurement of M. The calculated air concentration C_a is an estimate of suspended particulates. The height of the dust loading measurement specifies the height for the estimated air concentration.

Resuspension Factor

This model was used initially for estimating the resuspension of fallout particles deposited from atmospheric nuclear weapons tests. It is the earliest of the radioactivity resuspension models and was proposed by Langham[3] for relating airborne radioactivity concentrations above a contaminated area to measured soil surface radioactivity soon after deposition.

The initial formulation defined the resuspension factor S_f (length^{-1}) as the ratio of air concentration C_a (radioactivity/volume) to surface radioactivity C_s (radioactivity/area). Once calculated, S_f would predict air concentrations from measured surface radioactivity by

$$C_a = S_f C_s \qquad (3)$$

A major assumption underlying this calculation is that the air concentration is predictable from the local surface contamination. For this assumption to hold, it is necessary that the contaminated area be sufficiently large that an "equilibrium" air concentration results: the addition of more contaminated land upwind should not increase the measured air concentration. Care must be exercised in the measurement of the soil radioactivity, which is assumed to be a surface deposit.

This model does not account for any climatic or soil-related variables that could modify the resuspension phenomenon. These variables include such important factors as wind speed and soil moisture content, so this model and the other resuspension factor models that follow should be restricted to situations where a resuspension factor has been measured for the specific environmental conditions of interest. The calculated air concentration C_a, based on the resuspension factor, is an estimate of suspended particulates. The height at which C_a is estimated is determined by the height at which measurements were made in determining S_f.

Time-Dependent Resuspension Factor

Anspaugh et al.[3] discussed in some detail the development of a time-dependent resuspension factor $S_f(t)$ (length^{-1}) to account for decreased air concentrations over contaminated areas as the deposited particles become weathered into and intimately associated with the soil. The resulting time-dependent air concentration $C_a(t)$ (radioactivity/volume) is calculated by

$$C_a(t) = S_f(t)C_s \qquad (4)$$

where the soil radioactivity C_s (radioactivity/area), is assumed to remain constant with time, changing only in the degree of association with soil particles or the depth of mixing of the initial surface deposit.

Three models were referenced by Anspaugh et al.,[3] and all may be set in the form of Eq. 5. Additionally, a model of the time-dependent resuspension factor is used in the Uranium Dispersion and Dosimetry (UDAD) computer code.[5] This latter model has also been incorporated in the Final Generic Environmental Impact Statement on Uranium Milling (FGEIS)[6] and in the code MILDOS.[7] This model, too, fits the form

$$S_f(t) = S_f(0)\,e^{-\lambda f(t)} + S_f(\infty) \qquad (5)$$

where $S_f(0)$ = resuspension factor for a fresh deposit (length^{-1})

λ = weathering rate constant describing the decrease of resuspension per unit of the time function

$f(t)$ = function of time used

$S_f(\infty) = $ long-term asymptote that observed resuspension factors approach (length^{-1})

The time-dependent resuspension factor model shares the assumptions and limitations described for the original resuspension factor model.

Particle-Size-Dependent Resuspension Factor

The UDAD code[5] model of wind resuspension is both time and particle-size dependent. It is used for estimating resuspension over large areas downwind of uranium mining and milling facilities.

Momeni, Yuan, and Zielen[5] described the net vertical flux $F_v(z)$ (radioactivity/area-time) from soil to air (at height z) as

$$F_v(z) = RC_s - V_d C_a \qquad (6)$$

where the new parameters are resuspension rate R (time^{-1}) and the deposition velocity V_d (length/time). Under the steady-state assumption of large area uniform contamination, which has been noted as a requirement for the use of the resuspension factor, the net flux $F_v(z)$ will be zero because deposition will just equal resuspension.[5] So Eq. 6 may be manipulated to give

$$\frac{R}{V_d} = \frac{C_a}{C_s} = S_f \qquad (7)$$

Momeni, Yuan, and Zielen[5] assumed that the resuspension rate R is independent of particle size, while the deposition velocity V_d is particle-size dependent. Thus a particle-size-dependent resuspension factor is written as Eq. 8, where the subscript i is an index to particle size

$$S_{fi} = \frac{R}{V_{di}} \qquad (8)$$

If the resuspension factor and deposition velocity for any reference particle size are known, Eq. 9 allows the calculation of the resuspension factor for another particle size

$$S_{fi} = \frac{S_{fr} V_{dr}}{V_{di}} \qquad (9)$$

where the subscript r indicates the reference particle size.

The particle-size-dependent resuspension factor is also treated as time dependent in UDAD. The general equation has the form of Eq. 5, and substituting from Eq. 9

$$S_{fi}(t) = \frac{V_{dr}}{V_{di}} [S_{fr}(0) e^{-\lambda f(t)} + S_{fr}(\infty)] \qquad (10)$$

Resuspended air concentrations are given by the summation

$$C_a(t) = \sum_{i=1}^{n} C_{ai}(t) = \sum_{i=1}^{n} S_{fi}(t) C_{si} \qquad (11)$$

when $S_{fi}(t)$ is given by Eq. 10 and there are n particle-size classes.

This model, like the other resuspension factor models, is suitable for large area sources having reasonably uniform particulate deposition and is subject to the same basic assumptions and limitations.

Resuspension Ratio

Amato[8] developed a theoretical resuspension rate R_r consisting of the ratio of an air concentration due to resuspension of previously deposited particles C_a (radioactivity/volume) and an air concentration arriving directly from an active particulate source C_d (radioactivity/volume) as expressed by

$$R_r = \frac{C_a}{C_d} \qquad (12)$$

The total air concentration at any point is the sum

$$C_t = C_d + C_a = C_d (1 + R_r) \qquad (13)$$

Amato's development assumes a constant particulate release rate from the source, constant environmental conditions, and the downwind transport characterized by a Gaussian plume model. Under these conditions, the resuspension ratio was independent of the rate of contaminant release from the source.

The resuspension ratio as an estimator of resuspended particulates is intended for use in applications where (1) a history of particulate

Table 1 Environmental Variables and Model Parameters for the Wind Erosion Equation

Primary wind erosion variables	Equivalent erosion equation parameters
Soil erodibility index, I (function of soil particle size distribution; read from a table) Knoll erodibility, I_s (function of knoll slope steepness; read from a graph)	Soil and knoll erodibility, I' (the product of I and I_s)
Surface crust stability, F_s	Disregarded—crust is transient
Soil ridge roughness, K_r (function of height, width, and spacing of clods and furrows)	Soil ridge roughness factor, K' (estimated by comparison to a set of standard photographs)
Annual average wind velocity, v (read from map) Surface soil moisture, M [estimated using Thornthwaite's (Ref. 12) precipitation-evaporation index]	Local wind erosion climatic factor, C' (may be calculated but commonly read from maps of C')
Distance across field, D_f (field width in direction of primary erosive wind) Sheltered distance, D_b (calculated from barrier height upwind of field)	Field length, L' (the difference between D_f and D_b)
Quantity of vegetative cover, R' (mass of standing or fallen vegetative residue per unit area) Kind of vegetative cover, S (factor related to erosion-reducing effectiveness of residues from different crops) Orientation of vegetative cover, K_0 (factor relating erosion reduction to standing vs. fallen crop residues)	Equivalent vegetative cover, V (the product of R', S, and K_0)

release has resulted in a well-defined deposition pattern, (2) the particulate release rate is known, and (3) the ratio has been previously measured at locations of interest. This ratio is not as useful when the variation of environmental or source conditions is to be considered or when calculation of the release rate from a general particulate source is desired.

Wind Erosion Equation

Substantial research has been devoted to the development of a soil erosion equation by agricultural scientists.[9-11] Eleven primary variables have been determined to control the wind erodibility of land surfaces (Table 1). In developing a model of wind erosion, the 11 primary variables were embodied in the parameters indicated in Table 1. The potential annual average soil loss E (mass/area-time) is expressed as

$$E = f(I', C', K', L', V) \qquad (14)$$

where the parameters are given in Table 1. Since E is a complex function of the parameters, charts

and nomograms have generally been used to calculate the effect of each factor. Skidmore, Fisher, and Woodruff[13] and Fisher and Skidmore[14] have prepared a computer code to facilitate solving the equation.

The general relations between the primary variables, the equation parameters, and the potential soil loss E are summarized below according to the discussion in Woodruff and Siddoway[10] and Blackwood and Wachter.[15]

$$E \propto I' \propto I \cdot I_s \qquad (15)$$

$$E \propto C' \propto \frac{v^3}{M^2} \qquad (16)$$

$$E \propto K' \begin{cases} \propto K_r & (K_r > 3.5 \text{ in.}) \\ \propto \dfrac{1}{K_r} & (K_r < 3.5 \text{ in.}) \end{cases} \qquad (17)$$

$$E \propto L' \propto D_f - D_b \qquad (18)$$

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resuspension factor nt in UDAD. The Eq. 5, and substi-

$$^t) + S_{fr}(\infty)] \quad (10)$$

are given by the

$$; S_{fi}(t)C_{si} \qquad (11)$$

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an estimator of ...ended for use in ...ory of particulate

$$E \propto \frac{1}{V} \propto \frac{1}{R'SK_0} \qquad (19)$$

All of these parameters are described in Table 1.

The wind erosion equation is designed for use with small or large areas of land. Input parameters are generally annual average values and are comparable to the data generally available for generic environmental assessments. A great body of literature is available to give guidance in specific manipulations of the technique. The estimates given by the wind erosion equation are net soil loss (total mass/area-time), with no information regarding the respirable or suspended particle sizes. Some work of Gillette discussed later is of use in expressing the net erosion in terms of suspended dust flux.

It is difficult to present a concise summary of the wind erosion equation for two reasons. First, it is not a single equation; although it is based on information embodied in numerous theoretical and empirical relationships, no summary equation has been developed. Second, some relationships originally developed in terms of measured soil parameters have not been used directly in the final wind erosion equation. These have been replaced by relationships normalized to well-studied sites and expressed in terms of annual average values of more easily obtained quantities.

In later models reviewed here, some of the original parametric relationships have been picked up by other investigators and used to advantage. This is particularly true of Blackwood and Wachter[15] in the parametric emission rate; Gillette, Blifford, and Fenster[16] in the suspension flux measurements; and Travis[17] in the development of the combined suspension model. The fact that the relationships they employ do not appear explicitly above is due to the modifications made in generalizing the wind erosion equation.

Empirical Emission Factor

A number of authors have developed empirical emission factors for specific atmospheric pollution sources. Some examples are coal dust from coal storage piles;[15] rock dust from quarrying, processing, and handling rock products;[18] and dust from vehicular movements on dirt and gravel roads, in construction activities, and in agricultural activities.[19] Schwendiman et al.[20] used a similar technique in initial stages of evaluating dust releases from uranium mill tailings piles.

The emission factors are dust release rates Q (mass/time), estimated for a given source in its specific geometry. Emission factors are also found expressed in the units of a fractional release rate (mass released/mass of source-time), but this discussion will not use those units. Suspended (or respirable) dust concentrations $\chi(x,y,z;H)$ (mass/volume) are measured by receptors at several locations downwind from the source. The data are then fitted with a Gaussian plume atmospheric dispersion model to give the effective release rate (or emission factor)

$$Q = 2\pi\sigma_y\sigma_z\bar{u}\,\chi(x,y,z;H)\exp\left[\frac{y^2}{2\sigma_y^2}\right]$$

$$\times\left\{\exp\left[-\frac{(z-H)^2}{2\sigma_z^2}\right]\right.$$

$$\left.+ \exp\left[-\frac{(z+H)^2}{2\sigma_z^2}\right]\right\}^{-1} \qquad (20)$$

where this model represents a continuous point source release, and

where $x =$ downwind distance (length) from plume source to receptor

$y =$ crosswind distance (length) from the axis of the plume to receptor

$z =$ receptor height (length) above ground

$\bar{u} =$ mean wind speed (length/time) during the time of measurement

$H =$ effective release height (length) for the source

$\sigma_y =$ standard deviation in the crosswind direction of the plume concentration distribution

$\sigma_z =$ standard deviation in the vertical of the plume concentration distribution

For detailed discussions of various forms of such dispersion models, the reader is referred to Turner[21] or Slade.[22]

Although restricted to the conditions at the time of dust concentration measurements, an empirical emission factor gives an estimate of the magnitude of future releases from the specific source. The technique has been used typically to predict respirable particulate releases only and does not include any information on releases of saltating particles. With suitable dust concentration mea-

surements, the technique will estimate suspended dust releases as well. Calculations of empirical emission factors are required for each given application, and they are subject to the assumptions required for making Gaussian plume dispersion estimates.[21,22]

Parametric Emission Rate

Blackwood and Wachter[15] developed an analysis of coal dust emissions from coal storage piles. Although their conclusions were based on an empirical emission factor, a parametric relationship was developed to include some important environmental factors that influence particulate emissions. A parametric emission rate has the benefit of allowing a single calculated empirical emission factor to be used in a form that can be modified by changes in important environmental factors.

The emission rate Q (mass/time) is given the form

$$Q = \frac{K_e(u^a \rho^b s^c)}{(P-E)^d} \qquad (21)$$

where u = unobstructed wind speed (length/time) at a height equal to the mid-height of the coal storage pile

ρ = bulk density (mass/volume) of the pile

s = surface area (area) of the pile

$P-E$ = Thornthwaite's precipitation-evaporation index (dimensionless)[1]

The exponents a, b, c, and d are all empirical constants. Based on evaluations of data from soil erosion studies and wind tunnel coal dust entrainment studies, the most applicable ranges (and selected values)[15] for the four exponents are

$$2.7 \leqslant a \leqslant 3.0 \quad (a = 3)$$
$$2.0 \leqslant b \leqslant 5.9 \quad (b = 2)$$
$$c = 0.345$$
$$d = 2.0$$

The value of the proportionality constant K_e is calculated for each application by solving Eq. 21 for K_e and letting Q be the empirical emission factor determined when the other four parameters are known. Once K_e has been determined, Q may be calculated for different conditions of wind speed, pile area, pile density, and P-E index.

The parametric model of emission rate is of interest because it includes the effects of major physical factors and requires few measurements. As in the case of the empirical emission factor, this model estimates respirable or suspended particle releases with no treatment of saltation.

Saltation (Horizontal) Flux

Bagnold[23] studied the movement and drifting of dune sands in the Libyan Desert and developed wind tunnel techniques for measuring the horizontal flux of particles in saltation under varying conditions. Central to the discussion of saltation, and to the models presented later in this review, is a description of the wind-speed profile near the ground and the forces acting to lift particles from the earth into the airstream.

For the relatively high wind speeds needed to cause soil erosion, a plot of wind speed vs. the logarithm of height above the ground gives a straight line (Fig. 1). Different wind speeds, measured at
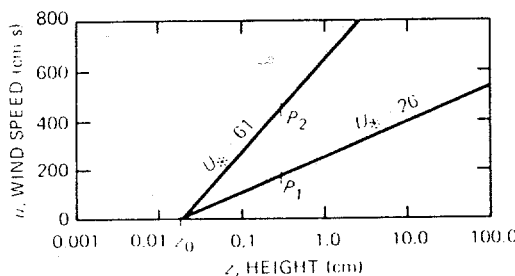

Fig. 1 Wind-speed profile: wind speed as a function of height above the ground (adapted from Ref. 23).

the same height, lie along separate wind-speed profiles as indicated by the points P_1 and P_2. Each wind-speed profile $u(z)$ (length/time) is defined by a common intercept z_0 and a unique slope. Each profile may be represented by

$$u(z) = \frac{1}{k} U_* \ln\left(\frac{z}{z_0}\right) \qquad (22)$$

where U_* is called the drag velocity or friction velocity and k is von Karmann's constant (= 0.4). The parameter z_0 is a small but measurable height above the ground at which the wind speed becomes zero. The parameter z_0 is dependent on the roughness features of the soil surface.

Figure 2 shows a sequence of wind profiles measured over a bed of sand. The three curves labeled U_* and intersecting at z_0 were measured
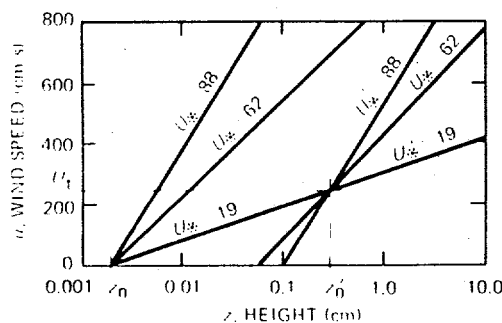
Fig. 2   Erosional wind-speed profile: U_* = without wind erosion, U_*' = with wind erosion (adapted from Ref. 23).

while the sand surface was stabilized against wind erosion by wetting. The three curves labeled by U_*' were measured while the surface of the sand was dry and subject to erosion. In the latter case, the amount of sand driven by the wind increased as U_*' increased. The zero wind-speed intercept was found at increasingly greater heights for greater U_*', as if the increased burden of saltating sand were acting as increased roughness of the physical surface. From Fig. 2 it is also apparent that, when erosion is in progress, the wind-speed profiles intersect at a common point. This point is used to define a height z_0' and a velocity U_t. Wind profiles during erosion may all be described in terms of the common point and a unique slope

$$u(z) = \frac{1}{k} U_*' \ln\left(\frac{z}{z_0'}\right) + U_t \qquad (23)$$

Manipulation of Eq. 23 yields a description of the friction velocity U_*' during wind erosion as

$$U_*' = \frac{k[u(z) - U_t]}{\ln(z/z_0')} \qquad (24)$$

Bagnold[23] defines three threshold velocities. The first, U_t, as used in Fig. 2 and Eqs. 23 and 24, is the impact threshold velocity. It is defined as the velocity U_t measured at z_0' for which saltation can just be indefinitely maintained by the impact of falling grains.

The fluid threshold velocity u_t describes the wind speed required to initiate particle movement by the wind, rather than that required to sustain the movement once it has started. This velocity is greater than the impact threshold and is expressed by

$$u_t = \frac{1}{k} U_{*t} \ln\left(\frac{z}{z_0}\right) \qquad (25)$$

where the threshold friction velocity U_{*t} is the friction velocity at which erosion begins.

By equating the forces of gravity and wind on an average particle on the surface of a sand bed, Bagnold gives an equation for the threshold friction velocity,

$$U_{*t} = A_f \left[\frac{gd(\sigma - \rho)}{\rho}\right]^{\frac{1}{2}} \qquad (26)$$

where A_f = an empirical constant (= 0.1)
 ρ = density of air
 σ = density of the grain
 g = acceleration of gravity
 d = grain diameter

Using Eq. 26, the fluid threshold velocity u_t may be rewritten

$$u_t(z) = \frac{A_f}{k} \left[\frac{gd(\sigma - \rho)}{\rho}\right]^{\frac{1}{2}} \ln\left(\frac{z}{z_0}\right) \qquad (27)$$

Equation 27 may be used to determine the wind speed required to initiate movement of a given particle size.

Bagnold[23] states that the impact threshold velocity U_t may also be described by a relationship in the form of Eq. 27 of the empirical constant A_f if replaced by another A_i, which is ~20% smaller (~0.08). If the height of measurement is specified as z_0', then

$$U_t = \frac{A_i}{k} \left[\frac{gd(\sigma - \rho)}{\rho}\right]^{\frac{1}{2}} \ln\left(\frac{z_0'}{z_0}\right) \qquad (28)$$

Measurements of the mass flux of saltation particles F_h [mass/length(crosswind)-time] allowed Bagnold to write a saltation model in terms of the friction velocity U_*'

$$F_h = C \left(\frac{d}{D}\right)^{\frac{1}{2}} \frac{\rho}{g} U_*'^3 \qquad (29)$$

where D is the reference grain diameter of 250 μm and C is an empirical constant equal to 2.8 for sand with a very wide range of particle sizes. Sub-

(25)

tuting Eq. 24 for U'_* gives the saltation flux in terms of the wind velocity measured at any height.

$$F_h = \frac{C\rho}{g}\left[\frac{d}{D}\right]^{1/2}\left\{\frac{k[u(z)-U_t]}{\ln(z/z'_0)}\right\}^3 \quad (30)$$

This model allows prediction of the saltation flux over a uniform sand bed but does not include a treatment of the suspension flux of fine dust that may be present in the bed. The model is suitable for an application such as a uranium mill tailings pile that is sandy in nature. It has been adapted for that purpose by Mills, Dahlman, and Olson[24] in a model discussed next. Bagnold[23] states that, in the presence of large particles, pebbles, rocks, or other immobile elements, these models may not hold.

Saltation-Driven Suspension

Mills, Dahlman, and Olson[24] developed a model of particulate suspension for uranium mill tailings based on the saltation models of Bagnold[23] and the suspension measurements of Gillette, Blifford, and Fenster.[16] Mills, Dahlman, and Olson used Eq. 30 directly from Bagnold[23] as a model of the saltation of uranium mill tailing sands. However, in the equation that Mills, Dahlman, and Olson presented for the threshold velocity U_t, required in Eq. 30, they have substituted Bagnold's fluid threshold velocity (Eq. 27) in place of the impact threshold velocity (Eq. 28) that should be used. Because the higher u_t is used in place of U_t, it is likely that this model will underestimate the rate of saltation at any given wind speed.

The saltation flux formulation as implemented by Mills, Dahlman, and Olson is composed of

$$F_h = \frac{C\rho}{g}\left[\frac{d}{D}\right]^{1/2}\left\{\frac{k[u(z)-u_t]}{\ln(z/z_0)}\right\}^3 \quad (31)$$

and

$$u_t = \frac{A_f}{k}\left[\frac{gd(\sigma-\rho)}{\rho}\right]^{1/2}\ln(z/z_0) \quad (32)$$

where the parameters are defined in the previous section. As noted, these equations differ from the Bagnold[23] formulations.

Dust suspension was estimated by Mills, Dahlman, and Olson using the data of Gillette, Blifford, and Fenster,[16] assuming the suspension flux F_v to be directly proportional to the saltation rate F_h as follows

$$F_v = KF_h \quad (33)$$

where K is a proportionality constant dependent on the height of the suspended dust measurement and conditions of the soil or tailings. Mills, Dahlman, and Olson[24] inferred a value for K of 10^{-5} m^{-1} for dusts 0.4 to 6.0 μm in diameter suspended at 3.75 m above agricultural soil. This value is derived from a single measurement presented by Gillette, Blifford, and Fenster.[16] More measurements are now available and should be considered. Because of the limited particle-size range represented by K, the vertical flux represents the respirable particles only. This model provides estimates of saltation and respirable particle flux but not the flux of all particles in suspension.

Suspension (Vertical) Flux

D. A. Gillette and other researchers of the U. S. National Center for Atmospheric Research prepared a series of reports describing particle-size measurements of suspended soil aerosols above eroding agricultural fields.[16,25-28] Several conclusions were supported by the measurements, which were limited to the range of particle diameters from 2 to 20 μm.

1. Suspension is not measured in the absence of saltation.
2. Size distributions of suspended particles match the particle-size distribution of the original soil.
3. Mass flux of suspended particles increases with wind speed.
4. Mass flux is greater over a soil with a larger mass fraction in the suspendible size range.

Based on his experimental data, Gillette[25] related suspension flux F_v (mass/area-time) to wind speed as

$$F_v = C_v\left[\frac{U'_*}{U_{*t}}\right]^{\gamma} \quad (34)$$

where $\gamma = >3$ and is highly soil specific
C_v = a proportionality constant
U'_* = drag velocity measured during erosion
U_{*t} = threshold drag velocity

Travis[29] fit the data of Gillette[25] and concluded that γ could be specified as

$$\gamma = \frac{P}{3} + 3 \qquad (35)$$

where P is the mass percentage of particles of diameter <20 μm in a given soil. The model developed by Travis[17,29] is discussed later. Gillette did not give a value for C_v; however, Travis[17] used the data to estimate a value of 2×10^{-10} g/(cm$^2 \cdot$s).

In comparing measured suspension and saltation fluxes, Gillette[25] found that the suspension flux increases more quickly with wind speed than does the saltation flux; this held for all soils tested. He reported a good fit of the saltation flux data to

$$F_h = C_h U_*'^2 (U_*' - U_{*t}) \qquad (36)$$

where C_h is a constant of proportionality. The data supported a value of $C_h = 1 \times 10^{-6}$ (g$\cdot$s^2/cm^4) for three soil types.

Combined Suspension Model

Travis presented a model describing the redistribution of wind-eroded soil and contaminant mixtures.[17,29] The initial portions of the model describe dust suspension as a function of saltation flux. The intended application was the assessment of air concentrations and the downwind spread of particulate ^{238}PuO$_2$ deposited as a result of a hypothetical nuclear heat-source accident. Parts of Travis' model were incorporated into the UDAD computer code by Momeni[5] and have become part of the FGEIS[6] and the computer code MILDOS.[7] In these documents the application is particulate releases from uranium mill tailings piles. The version of the model incorporated into UDAD, MILDOS, and FGEIS is somewhat simplified.

Travis[17] follows Gillette, Blifford, and Fenster[16] in using a simplified form of the wind erosion equation to estimate the saltation flux of soil F_h. A simple erodibility function X [mass/length(crosswind)-time], based on early versions of the wind erosion equation, is used:

$$X = \frac{aI}{(RK_r)^b} \qquad (37)$$

where I is the soil erodibility index based on the fraction of the soil mass in particles >840 μm, R is the amount of vegetative residue (mass/area), K_r is a surface roughness equivalent (length), and two empirical constants are represented by a and b Gillette, Blifford, and Fenster[16] proposed correcting this function for wind speed

$$F_h = X \left[\frac{U_*'}{U_{*r}} \right]^3 \qquad (38)$$

where U_{*r} is a reference friction velocity for the conditions when the formulation for X was determined.

The equation used in Travis' model[17] is further modified by using the form suggested by Gillette[25] (Eq. 36) and using an effective friction velocity U_{*e} that accounts for soil moisture and the presence of large roughness elements, which may reduce the friction velocity acting on the soil. The large roughness elements might be large rocks, plants, or trees; they affect soil erosion in addition to the effect of soil roughness features accounted for by K_r in Eq. 37. Thus Travis predicts horizontal soil mass flux using

$$F_h = X \left[\frac{U_{*e}}{U_{*r}} \right]^2 \left[\frac{U_{*e}}{U_{*r}} - \frac{U_{*t}}{U_{*r}} \right] \qquad (39)$$

This description of F_h is not used in UDAD, MILDOS, or FGEIS. In the versions implemented in those computer codes, the horizontal mass flux F_h is calculated from Eq. 36. A modification is applied to account for soil moisture content, but vegetative cover, soil surface roughness, and large roughness elements are not considered in UDAD, MILDOS, or FGEIS.

In developing a description of the vertical mass flux of soil, F_v, Travis[17] started with a general relationship for saltation,

$$F_h = C_h U_*'^3 \qquad (40)$$

where C_h is the constant used in Eq. 36. The vertical flux of particles, F_v, is taken by Travis from Gillette[25] as

$$F_v = C_v \left[\frac{U_*'}{U_{*t}} \right]^{(P/3 + 3)} \qquad (41)$$

where the parameters are as previously defined. Travis combines Eqs. 40 and 41 using the identity $F_h/C_h U_*^3 = 1$, giving

$$F_v = \left[\frac{F_h C_v}{C_h U_{*t}^3}\right]\left[\frac{U_*'}{U_{*t}}\right]^{P/3} \qquad (42)$$

Travis notes that as U_*' approaches U_{*t} the suspension flux must go to zero, and he forces this limit by subtracting 1

$$F_v = F_h\left[\frac{C_v}{C_h U_{*t}^3}\right]\left[\left(\frac{U_*'}{U_{*t}}\right)^{P/3} - 1\right] \qquad (43)$$

This formulation has several other important features. First, as the mass percentage of particles <20 μm approaches zero, the suspension flux is forced to zero. Second, the suspension flux has been made functionally dependent on the saltation flux F_h, which Travis calculates from Eq. 39. Third, the calculated F_h is normalized by the ratio of suspended to saltating mass (C_v/C_h) observed by Gillette.[25] These modifications make the vertical flux model quite interesting, and it has been incorporated directly into UDAD, MILDOS, and FGEIS.

Concentration Gradient

Shinn et al.[30] measured dust concentrations as a function of height and wind speed at two sites. They found that the concentrations followed a power function of height. Because the measurements were restricted to particles <10 μm in diameter, this model is useful only for respirable particle sizes.

The dust concentration power function may be written in concentration gradient form as

$$\frac{d\chi(z)}{dz} = \frac{p\chi(z)}{z} \qquad (44)$$

where $\chi(z)$ (mass/volume) is the airborne dust concentration at height z and p is an empirical constant (~ 0.3). The vertical mass flux of dust, $F_v(z)$ (mass/area-time), in terms of the concentration gradient is

$$F_v(z) = \frac{\kappa \, d\chi(z)}{dz} \qquad (45)$$

where κ is the eddy diffusivity, a measure of the turbulent mixing of the air. It is expressed as

$$\kappa = kz U_*' \qquad (46)$$

where k is von Karmann's constant ($= 0.4$) and U_*' is the friction velocity. Combining Eqs. 44 to 46 gives

$$F_v(z) = pkU_*'\chi(z) \qquad (47)$$

Shinn et al.[30] restricted their flux values to 1-m height, noting that only a 20% error is incurred so long as the concentrations are measured between 0.7 and 2.0 m. This restriction simplifies the flux expression

$$F_v(1) = pkU_*'\chi(1) \qquad (48)$$

The concentration data of Shinn et al. for two sites fit the form

$$\chi(1) = C_c U_*'^{\alpha} \qquad (49)$$

where C_c is a proportionality constant and α is >2. Both C_c and α appear to be very soil specific. Incorporating Eq. 49 into Eq. 48 gives

$$F_v(1) = pkC_c U_*'^{\gamma} \qquad (50)$$

where γ is $\alpha + 1$. We use the symbol γ here purposely to recall the formulation presented by Gillette, Eq. 34, which is similar. The form of the equations differ; thus Gillette's C_v is not identical to Shinn's pkC_c, but they are clearly related.

Shinn further developed the similarity by solving Eq. 50 for pkC_c, stating that measurement of the flux and friction velocity for a known case, denoted as $F_0(1)$ and U_{*0}, determined C_c for the particular site. Substituting the measured values then gives

$$F_v(1) = F_0(1)\left[\frac{U_*}{U_{*0}}\right]^{\gamma} \qquad (51)$$

by which the flux at any friction velocity is determined for the specific site. If the U_{*0} is allowed to

be the threshold friction velocity U_{*t}, then correspondence in form is achieved with Eq. 34. Thus Gillette's C_v corresponds to the flux at U_{*t}, with the difference that Gillette's measurements were made at $z = 3.75$ m.

Resuspension Rate

Healy[31] developed a model to provide estimates of radioactivity concentrations in air over an area of deposited radioactive particles. The original application was to radioactive particles deposited at the Nevada Test Site as a result of nuclear weapons tests.

Healy's model uses a numerical technique to account for source size and geometry, but the technique is not presented in available reports. The estimated airborne radioactivity is used as input to a Gaussian plume atmospheric dispersion model. The airborne radioactivity estimates are based on a resuspension rate R (time^{-1}) and the radioactivity on the soil C_s (radioactivity/area). The product of these parameters is equivalent to the radioactivity flux, according to Anspaugh et al.[3]

$$F_v = RC_s \tag{52}$$

For this relationship to hold, the radioactive contaminant on the soil surface must be measured in terms of its presence in the suspendible fraction of the surface soil. In contrast to Eq. 6, the effect of deposition has not been included, and the model holds for particles of negligible deposition velocity.

Anspaugh et al.[3] further pursued the relations between the resuspension rate R (time^{-1}), the airborne dust concentration χ (mass/volume), and the resuspension factor S_f (length^{-1}).

$$R = \frac{-pkU_* \chi}{C_s} = -pkU'_* S_f \tag{53}$$

where k is von Karmann's constant, U'_* is the friction velocity (length/time), and p is the power of a power function describing dust concentration vs. height. This formulation is based on the work of Shinn et al.[30] described previously. The dust concentration and flux values were restricted by assumption to 1-m height, and the relationships were based on measurements limited to the respirable particle-size range.

CATEGORIZATION OF MODELS

The number and diversity of models reviewed makes a concise summary and categorization desirable. Table 2 is an attempt to provide both a quick reference and a useful tabulation of the models' features. In Table 2 three major subdivisions are evident, and these are described further in the following paragraphs.

Suitable Applications

The two categories under this heading indicate the conditions for which the model was developed, or for which it is most suitable because of simplifying assumptions or explicit parameterization.

Source Size. Some models assume an infinite source, whereas others are developed specifically for small or limited areas. No estimate is made of the cutoff between small and large for model application. However, those models requiring large areas generally depend on the assumption that the upwind expanse is sufficient to produce an equilibrium air concentration. In other words, the addition of more contaminated land upwind should not increase the measured air concentration.

Material. Three source material categories are offered to describe the intended applications. "Deposited particles" implies applications where the particles of interest are not of the same nature as the soil or other substrate. "Soil-like sources" identifies models that (1) are applicable to the movement of soil particles and (2) describe the movement of contaminants intimately mixed with soil. "Specific application" is a category for models that apply only to one specific source type or geometry.

Environmental Parameters

The eight categories included in Table 2 relate to features that enhance the representation of source or environmental conditions. Because wind suspension is in fact a function of these parameters and others, it may be assumed that lack of explicit parameterization implies simplifying assumptions used to limit dependence on the parameter or that measured input data implicitly include the effects of environmental conditions at the time of measurement.

Table 2   Summary of Entrainment Model Categorization

Model	Deposited particles	Soil-like sources	Specific application	Small area	Large area	Wind speed	Source particle size	Source size	Surface roughness	Moisture content	Weathering of source	Vegetative cover	Large roughness elements	Radioactivity/volume (susp.)	Mass/area-time (net)	Mass/time (resp.)	Mass/length-time (net)	Mass/area-time (2-20 μm dia.)	Mass/area-time (resp.)	Radioactivity/area-time (resp.)
Mass loading	X			X	X									X						
Resuspension factor	X			X	X									X						
Time-dependent resuspension factor	X			X	X						X			X						
Particle-size-dependent resuspension factor	X			X	X		X				X			X						
Resuspension ratio				X	X									X						
Wind erosion equation		X		X		X	X	X	X	X		X	X							
Empirical emission factor		X	X		X	X	X	X					X		X					
Parametric emission rate		X	X			X	X		X	X		X	X			X				
Saltation flux		X	X			X	X			X						X		X		
Saltation-driven suspension		X				X	X		X								X	X		
Suspension flux		X				X	X		X			X						X		
Combined suspension model		X		X		X	X		X	X		X	X							
UDAD/MILDOS/FGEIS modification of combined suspension model		X		X		X			X	X								X	X	
Concentration gradient	X				X	X	X													
Resuspension rate					X	X	X													X

Quantity Estimated

Seven categories are used to group the model estimates in terms of their dimensions and particle-size restrictions. Generic dimensions are given in Table 2, but each model's constants or measured parameters may impose a specific system of units. Particle-size restrictions are noted parenthetically with abbreviations for respirable (resp.), suspended (susp.), net loss (net), or with specific particle-size ranges in terms of particle diameter (dia.). These size classes are somewhat flexible, and specifics should be consulted in this text and the original reports.

CONCLUSION

Environmental impact assessments and interest in environmental cycling of radionuclides have produced needs for models of particle entrainment, dispersion, and deposition. In this review we focus on the commonly available models for saltation, suspension, and resuspension. Our purpose is twofold: (1) to present in one place a brief discussion of the several techniques in use and (2) to categorize in an outline fashion the essential features of the models.

Shaeffer[32] presents an approach to model evaluation that is useful in putting this review in perspective. He divides the methodology into six major tasks: (1) model examination, (2) algorithm examination, (3) data evaluation, (4) sensitivity analyses, (5) validation studies, and (6) code comparisons. The first five items are essential in the development of an acceptable model, and the sixth is limited to cases where computer codes exist. This article addresses only item 1 of Shaeffer's list; the burden of fully evaluating a model for a specific use falls to the individual researchers.

Further investigation of these models will reveal that they all are greatly data limited. The measurements supporting the more simple resuspension factor and mass loading models are prevalent, and this results in their general dominance in many uses. Certain benefits are to be gained, however, from judicious application of the more mechanistic models that incorporate the effects of major environmental factors. These benefits include the ability to evaluate the uncertainty associated with estimates resulting from reasonable variability assumed for each of the environmental parameters. In many cases the ability to set justifiable limits on

the variability of an estimate may be as important as the estimate itself.

Another benefit may arise from the concurrent evaluation of simple and complex models. Available data may support the use of a simple model, but evaluation of the same problem with a mechanistic model may lead to increased understanding. Situations where changes over time are apparent may be especially well suited for study with the more mechanistic models. Cases where mitigating actions might be considered for reducing particulate releases might well be studied in a time-dependent fashion using a model that can reflect the temporal variation of important environmental factors.

Validation of entrainment models has been grossly inadequate. Models have been formed on the basis of limited data and have often received extensive use without the benefit of subsequent validation or verification. A case of interest is the use of portions of Travis' Combined Suspension Model in MILDOS, UDAD, and FGEIS. This model was developed from work of diverse origins, is quite general, and is exceptionally parametric. It was adopted and has received use of considerable impact in these three computer codes. Yet, to our knowledge, it has never received corroborative verification or controlled validation.

We encourage researchers to carefully consider not only the assumptions and inherent limitations underlying the formulation of the models used but also to give considerable attention to the depth and quality of data available as input. For those involved in data collection and entrainment measurements, we hope that this article may generate some interest in concomitant measurement of some important environmental factors needed to allow validation of the models.

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